

ENVIRONMENTAL RESEARCH
LETTERS

TOPICAL REVIEW

OPEN ACCESS

RECEIVED

9 November 2020

REVISED

24 February 2021

ACCEPTED FOR PUBLICATION

9 March 2021

PUBLISHED

24 March 2021

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A review of indirect N₂O emission factors from artificial
agricultural watersJackie R Webb^{1,*} , Tim J Clough² and Wendy C Quayle¹¹ Centre for Regional and Rural Futures (CeRRF), Deakin University, 458 Research Station Road, Griffith, NSW 2680, Australia² Agriculture & Life Sciences Division, Lincoln University, PO Box 84, Canterbury, New Zealand

* Author to whom any correspondence should be addressed.

E-mail: j.webb@deakin.edu.au**Keywords:** artificial aquatic ecosystems, nitrous oxide, drainage, irrigation, ditches, ponds, channelsSupplementary material for this article is available [online](#)

Abstract

Nitrous oxide (N₂O) produced from dissolved nitrogen (N) compounds in agricultural runoff water must be accounted for when reporting N₂O budgets from agricultural industries. Constructed ('artificial') water bodies within the farm landscape are the first aquatic systems that receive field N losses, yet emission accounting for these systems remains under-represented in Intergovernmental Panel on Climate Change (IPCC) emission factor (EF) guidelines and global N₂O budgets. Here, we examine the role of artificial waters as indirect sources of agricultural N₂O emissions, identify research gaps, and explore the challenge of predicting these emissions using default EFs. Data from 52 studies reporting dissolved N₂O, nitrate (NO₃), and EFs were synthesised from the literature and classified into four water groups; subsurface drains, surface drains, irrigation canals, and farm dams. N₂O concentration varied significantly between artificial waters while NO₃ did not, suggesting functional differences in the way artificial waters respond to anthropogenic N loading. EFs for the N₂O–N:NO₃–N concentration ratio were highly skewed and varied up to three orders of magnitude, ranged 0.005%–2.6%, 0.02%–4.4%, 0.03%–1.33%, and 0.04%–0.46% in subsurface drains, surface drains, irrigation canals, and farm dams, respectively. N₂O displayed a non-linear relationship with NO₃, where EF decreased exponentially with increasing NO₃, demonstrating the inappropriateness of the stationary EF model. We show that the current IPCC EF model tends to overestimate N₂O production in response to NO₃ loading across most artificial waters, particularly for farm dams. Given their widespread existence, there is a need to: (a) constrain their global abundance and distribution; (b) include artificial waters in the global N₂O budget, and (c) expand the study of N processing in artificial waters across a geographically diverse area to develop our biogeochemical understanding to the level that has been achieved for rivers and lakes.

1. Introduction

Anthropogenic inputs of nitrogen (N) have increased nitrous oxide (N₂O) emissions from the world's river networks by four-fold during the 21st century (Yao *et al* 2020). Globally, agriculture contributes ~60% (4.3 Tg N yr⁻¹) of anthropogenic N₂O emissions (7.3 Tg N yr⁻¹), a result of increasing manure inputs within the pastoral sector and the widespread use of nitrogenous fertilisers on arable land (Tian *et al* 2020). Agriculture contributes to N₂O emissions as a result of the direct production of

N₂O within field soils (2.3 Tg N yr⁻¹), direct emissions from animal waste and manure management (1.5 Tg N yr⁻¹), and from N₂O produced from inland waters (0.5 Tg N yr⁻¹) due to increased N loading from agricultural leaching and runoff (Tian *et al* 2020). The incidental emissions from aquatic ecosystems, termed 'indirect emissions', are recognised as a key component of agricultural N₂O inventories, due to ~24% of terrestrial agricultural N inputs being lost via hydrological pathways (Hergoualc'h *et al* 2019). A large proportion of indirect emissions are generated in water bodies located in close proximity

or connected to agricultural fields, with agricultural drainage waters responsible for 20%–50% of direct N_2O emissions in some agricultural watersheds (Fu *et al* 2018, Billen *et al* 2020).

Emission factors (EFs) applied to dissolved inorganic N measurements are used to estimate N_2O emissions from inland waters where local measurements are lacking. They are used for reporting in both international (i.e. UNFCCC), national, and regional scale contexts to provide bottom-up estimates of greenhouse gas inventories. The Intergovernmental Panel on Climate Change (IPCC) provides guidelines on default EFs for waters contributing to indirect N_2O emissions from leaching and runoff (EF_5), which is further subdivided into groundwater and surface drainage (EF_{5g}), rivers and reservoirs (EF_{5r}), and estuaries (EF_{5e}). EFs can be calculated via two different methods using (a) the ratio of the N_2O flux to total N input (EF_A), or (b) the ratio of dissolved N_2O –N concentration to nitrate (NO_3)–N concentration (EF_B). Both approaches rely on estimates of N loading to water bodies, which are in turn subject to uncertainties. For aquatic N_2O emissions, previous past assessments to predict global N_2O emissions have focused on rivers, reservoirs, and estuaries (Seitzinger *et al* 2000, Maavara *et al* 2019, Tian *et al* 2020), with little attention paid specifically to artificial surface waters.

The most appropriate EF for each water group has been the centre of controversy for decades, with upwards and downwards revision as more data becomes available. Indirect N_2O EFs from a broad range of water types, including natural and human-made systems, are generalised into the two main groups: EF_{5g} and EF_{5r} . Based on a recent revision which derived concentration data from 192 surface waters, the IPCC prescribes a fixed EF value of 0.6% and 0.26% for EF_{5g} and EF_{5r} , respectively (Hergoualc’h *et al* 2019). This approach assumes a linear increase in N_2O with NO_3 loading, along with assumptions about the rates of N_2O produced from denitrification and nitrification. The range in EFs vary by orders of magnitude for both drainage waters (0.01%–18.4%) and rivers (<0.01%–2.75%, Hergoualc’h *et al* 2019). Poor quantification of indirect N_2O emissions is limited by the lack of studies that incorporate different types of agricultural surface waters (Outram and Hiscock 2012, Xia *et al* 2013, Xiao *et al* 2019), unclear definitions of water types, and inconsistent reporting of factors other than NO_3 that can influence N_2O production. Lack of definition for agricultural surface waters assumes that the mechanisms leading to N_2O supersaturation to be similar between groundwater and surface drainage (N_2O source from groundwater), and between rivers, lakes, and reservoirs (N_2O produced *in situ* by nitrification/denitrification). Importantly, current indirect EFs provided by the IPCC do not distinguish between agricultural artificial surface waters, hereafter referred to as ‘artificial waters’.

Artificial aquatic ecosystems are designer ecosystems constructed for human purposes, and may be wholly engineered systems where no previous water body existed or physically modified natural waters (i.e. channelized stream) (Clifford and Heffernan 2018). In agricultural landscapes, these artificial waters include channels constructed for subsurface and surface drainage, irrigation canals, and on-farm storage reservoirs (farm dams). These systems are designed to facilitate agricultural production and have human-defined hydrology. Some studies have demonstrated significant differences in dissolved N_2O concentrations between natural and agricultural water bodies, concluding that the application of a single EF value is inappropriate for upscaling indirect aquatic emissions (Outram and Hiscock 2012, Xiao *et al* 2019). Although there has been an increase in studies measuring N_2O and N loading in artificial waters, there remains a lack of quantitative understanding to guide predictions of indirect N_2O emissions from these systems.

Currently, IPCC EFs for estimating indirect N_2O emissions do not distinguish between artificial waters and natural surface waters (rivers, streams, and lakes) in agricultural landscapes. Agricultural surface waters such as drains, ditches, and canals are assigned the same EF as groundwater and headwater streams, yet represent 23% of data determining the default emission value of EF_{5g} (Tian *et al* 2019). Farm dams and other small farm-scale impoundments fall under EF_{5r} for rivers, lakes, and reservoirs, yet only represent ~7% of this category (Tian *et al* 2019). This review aims to address these gaps by: (a) explicitly assessing the role of different artificial agricultural waters as sources of indirect agricultural N_2O emissions; and (b) quantitatively evaluating their response to NO_3 loading in the context of the current IPCC indirect N_2O EF estimates. We draw on available data from the published literature on artificial waters and review the knowledge of N_2O production for each water type. We anticipate the findings in this review will lead to greater recognition, improved definition, and eventually, explicit inclusion of artificial waters in agricultural N_2O budgets to help account for uncertainties in agricultural N_2O accounting.

2. Methods

2.1. Literature search and classification

We employed a similar approach to that of Tian *et al* (2019) in our literature search for studies reporting dissolved N_2O and NO_3 concentrations in agricultural surface waters. The inclusion criteria was redefined to only include constructed, engineered, or modified water types located directly within the agricultural landscape (i.e. no rivers, natural creeks, or hydroelectric reservoirs). Data for EFs were gathered from studies included in the most recent IPCC update on EF_{5g} and EF_{5r} (Tian *et al* 2019), and a

Table 1. Types of artificial or modified agricultural surface waters and their description considered in this review.

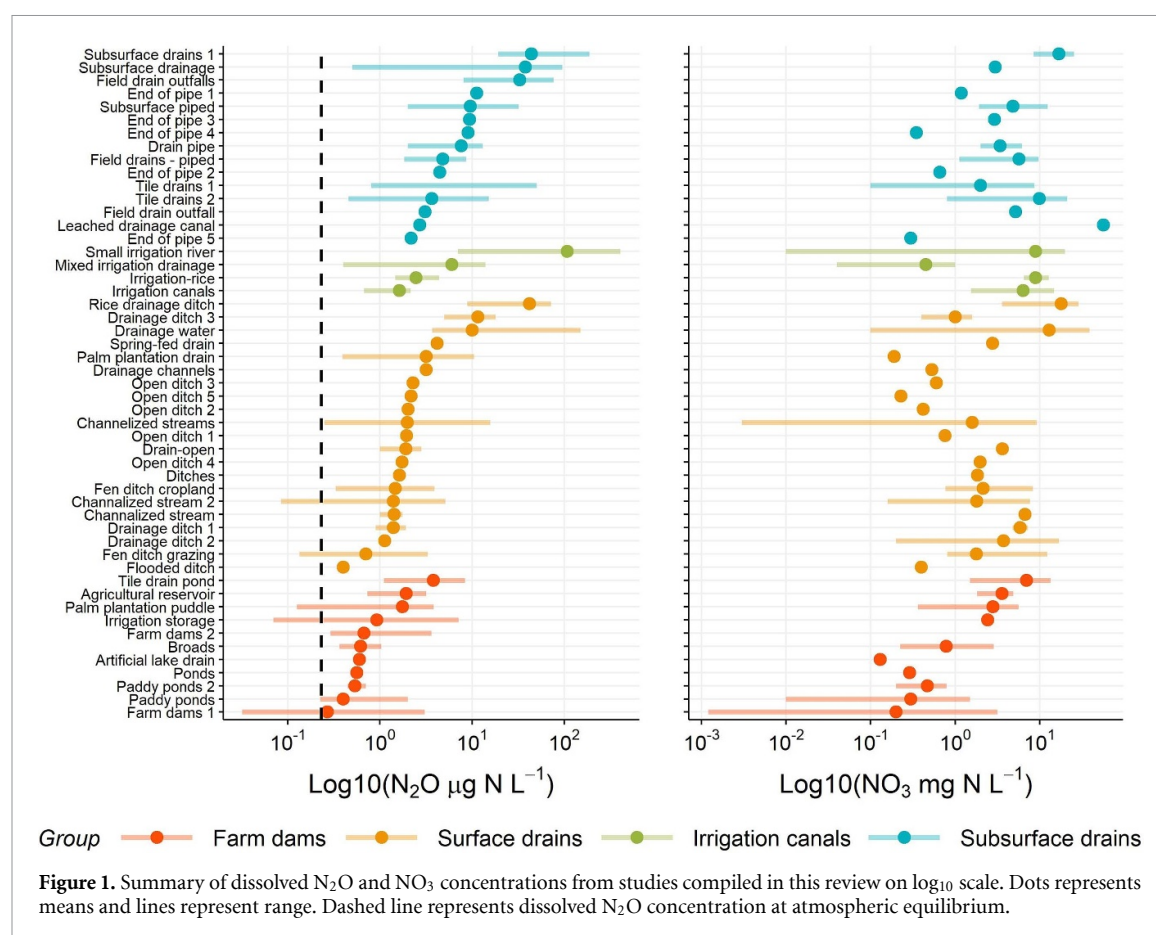
Agricultural surface water type	Description	Water types included
Subsurface drains	Subsurface drainage or other drainage structures not exposed to the atmosphere. High N_2O concentrations can accumulate and result in rapid degassing to atmosphere at field outlets.	Tile drains, piped drains, tile drain outlets.
Surface drains	Drainage channels for transporting excess agricultural water off the landscape. These structures are exposed to the atmosphere. Viewed as ‘zero-order’ streams.	Drainage ditches, drainage canals, drainage channels.
Canals	Canals and channels transporting irrigation water. Used for both water supply and removal on irrigated fields.	Any canal or channel supplying or removing water on irrigation land.
On-farm dams	Small waterbodies ($<10\,000\text{ m}^2$ area) of open water located directly within the agricultural landscape. Used for water supply, water storage, recreation, and flood control. Aquaculture ponds not included.	Artificial/modified wetlands, treatment wetlands (e.g. dairy effluent), rice paddy wetlands, livestock watering dams, irrigation storage dams, small reservoirs (not used for energy production).

Google Scholar search was conducted using the key terms ‘ EF_5 ’ or ‘dissolved N_2O concentrations’ combined with water definitions (‘wetland’, ‘pond’, ‘dam’, ‘reservoir’, ‘drainage’, ‘storage’, ‘canal’, ‘ditch’, ‘tile drain’, ‘channel’) under the theme ‘agriculture’. To be included, studies needed to (a) adequately define the water type to ensure it was an artificial or hydrologically modified system; (b) be located within an agriculturally dominant catchment; and (c) have an EF calculated from the N_2O – N/NO_3 – N concentration ratio. We chose to focus on the N_2O – N/NO_3 – N mass ratio method for calculating EFs as most studies lack the detailed mass balance information of N transport and N_2O flux measurement in their respective catchments required for EF_A . Mean values from both temporal and spatial datasets were included if they reported N_2O and NO_3 concentrations. We opted not to include aquaculture ponds and flooded rice paddy waters as N_2O emissions from these systems would be considered direct agricultural emissions. Following this criteria, an additional 25 studies were collected on top of the 27 artificial water studies included in the current IPCC EF estimates (Tian *et al* 2019).

Table 1 describes the classification of agricultural surface waters considered in this review. Subsurface drains are drainage structures installed directly under agricultural fields to control groundwater levels and prevent waterlogging (Waller and Yitayew 2015). Originally known as tile drains, these structures often consist of a dense network of perforated pipes placed 1–2 m below the surface and which discharge into open collection drains. The concentration of nutrients is highest in these waters as they increase soil water infiltration and collect leached water directly from the soil surface. Surface

drains are open waterways, such as ditches, that artificially remove excess water off the farm landscape. They include both on-farm zero-order field drains, main collection drains, and outlets. This infrastructure exists in flood prone areas such as wetlands, floodplains, and poorly draining soils, and acts to enhance agricultural productivity through improved efficiency, timing, and variety of farming operations (Christen *et al* 2001, Herzon and Helenius 2008). Canals represent engineered channel structures that can act as either supply or drainage channels for irrigated agricultural fields. These networks are often constructed in dryland agricultural regions and support intensive irrigated broad acre production of row crops and horticulture. Although drainage ditches and irrigation canals are structurally similar, being linear channels transporting flowing water, we decided to make a distinction here, largely because they represent inherently different farming systems (e.g. ditches located on farms where excess water is a limitation and canals located in dry regions where water needs to be artificially supplied).

On-farm dams are an integral part of the agricultural landscape that exist to store, control, recycle, and treat water on the farm. Here, we include all constructed waterbodies located directly in the farming landscape, including ponds and wetlands that have been modified or impacted by agriculture beyond their natural state. They exist across all areas of agriculture, including arable to livestock land and intensive commercial scale to small landholder scale (Chumchal *et al* 2016). In terms of nutrient cycling these water bodies behave functionally differently to drainage waters (as discussed below) and have been suggested as a management strategy to reduce surface N runoff (Siegfried *et al* 1994, Passy *et al* 2012).



2.2. Analysis

Data were categorised into four water groups: sub-surface drains, surface drains, canals, and farm dams (table 1). To assess significant differences between water groups, means of N_2O , NO_3 , and EFs were analysed using a one-way analysis of variance followed by a Tukey post-hoc test ($p < 0.05$) performed in R version 3.6.3 (R Core Team 2020). N_2O , NO_3 , and EF data were \log_{10} -transformed prior to analysis after checking for skewness using the skewness function in R package e1071 (v1.7–3; Meyer *et al* 2019). EFs were converted to a percentage for plotting and represent the percentage of $\text{N}_2\text{O-N}$ mg l^{-1} relative to $\text{NO}_3\text{-N}$ mg l^{-1} present in the water.

To test the relationship between N_2O and NO_3 , a Pearson's least square regression for linear covariance ($p < 0.05$), was applied to \log_{10} -transformed data and assessed by water group. Models were compared with IPCC predictions by applying EF_{5g} and EF_{5r} (0.0060 and 0.0026) to NO_3 data collected in this review, and which assumes a linear increase in N_2O concentrations with increasing NO_3 . Other environmental drivers that may be of importance in predicting aquatic N_2O emissions were also tested, given data availability. Variables included dissolved organic carbon (DOC), pH, annual N fertiliser applied, and precipitation.

Finally, artificial waters were compared with natural waters to assess whether artificial waters behave

differently as sources of indirect N_2O emissions. Here, N_2O , NO_3 and EF data for natural waters collected in Tian *et al* (2019), for the 2019 IPCC refinement, were used, and included waters defined as groundwater, streams, rivers, and lakes in agricultural catchments.

3. Results

A total of 52 studies met the selection criteria for artificial waters defined in this review, which included 15 sub-surface drains, 22 surface drains, 11 farm dams, and 4 irrigation canals. Over half of studies were located in Europe, followed by 30% in Eastern to South-Eastern Asia, 13% in North America, and 8% in the Southern Hemisphere (figure S1 (available online at stacks.iop.org/ERL/16/043005/mmedia)). These included land uses such as arable systems which were mostly annual row-crops ($n = 19$), pasture grazing for dairy and animal production ($n = 18$), rice ($n = 6$), irrigated land ($n = 3$), palm oil plantations ($n = 2$), and a mix of cropping and pasture ($n = 3$).

Mean N_2O , NO_3 concentrations, and EFs across all studies ranged $0.27\text{--}108$ $\mu\text{g N}_2\text{O-N l}^{-1}$, $0.13\text{--}57$ $\text{mg NO}_3\text{-N l}^{-1}$, and $0.005\%\text{--}4.37\%$, respectively (table S1, figure 1). In five studies, undersaturated N_2O concentrations relative to atmospheric equilibrium were reported, indicating potential negative emissions (figure 1). Within study variation

was large, with up to one and two orders of magnitude between the minimum and maximum recorded values for N_2O and NO_3 concentrations, respectively. The largest variation observed was in drainage water from a heavy clay soil (Dowdell *et al* 1979), a small irrigation river for rice (Hasegawa *et al* 2000), and subsurface drainage from an onion field (Sawamoto *et al* 2003). This variation was attributed primarily to seasonal changes. For EFs, within study variation was as low as 0.04% in agricultural reservoirs holding water for irrigation (Wang *et al* 2017) and as high as 38% in a spatial survey of farm dams (Webb *et al* 2019a).

3.1. Differences between water groups

Significant differences between artificial waters were only observed for N_2O concentrations (figure 2). Sub-surface drains, surface drains, and farm dams were significantly different from each other with means of 13.04, 6.16, and $1.09 \mu\text{g N l}^{-1}$, respectively for N_2O (table S1). Irrigation canals were significantly different from farm dams and had the highest mean N_2O concentration of $29.52 \mu\text{g N l}^{-1}$, although this value was highly skewed by one high value ($108 \mu\text{g N l}^{-1}$). In contrast to N_2O concentrations, NO_3 and EFs did not vary significantly with water group. The highest mean NO_3 concentration was recorded in a sub-surface drainage site of a vegetable farm, in a region with high annual rainfall (Saitama Prefecture, Japan), while the lowest mean concentration was observed in an artificial lake collecting subsurface and surface drain discharge from dairy farms. In contrast, the sub-surface drainage site with the highest NO_3 loading reported the lowest mean EF of 0.005% (Hasegawa *et al* 2000).

3.2. N_2O and EF relationship with NO_3

Across all artificial waters, significant linear relationships for both log-transformed N_2O concentrations ($R^2 = 0.22$, $p < 0.05$) and EFs ($R^2 = 0.29$, $p < 0.05$) were observed with log-transformed NO_3 concentrations (figure 3). The rate of increase in N_2O concentrations with increasing NO_3 concentration was lower than that predicted by the IPCC EF_{5g} and EF_{5r} models (figure 3(A)). EFs demonstrated an inverse relationship with increasing NO_3 loading (figure 3(B)). When considering water groups separately, different responses to NO_3 were shown (figure S2), with only farm dams found to have a significant positive correlation between N_2O and NO_3 ($R^2 = 0.79$, $p < 0.05$). Subsurface drains, surface drains, and farm dams all had significant inverse relationships between EF and NO_3 (figure S3). Relationships were difficult to assess for irrigation canals due to the lack of observations ($n = 4$).

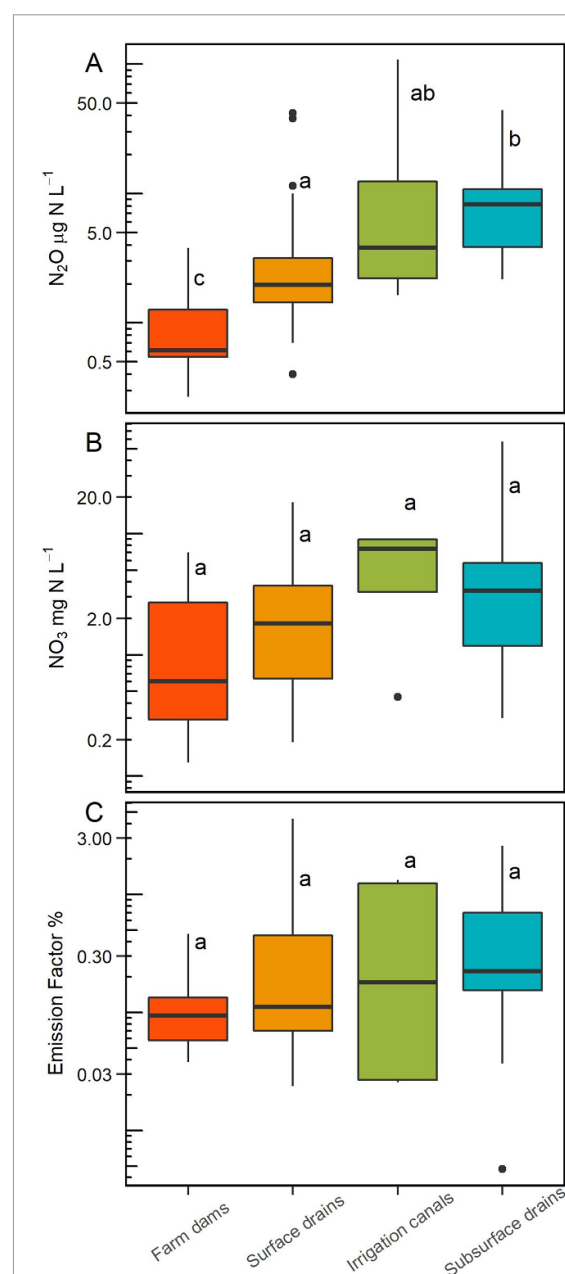
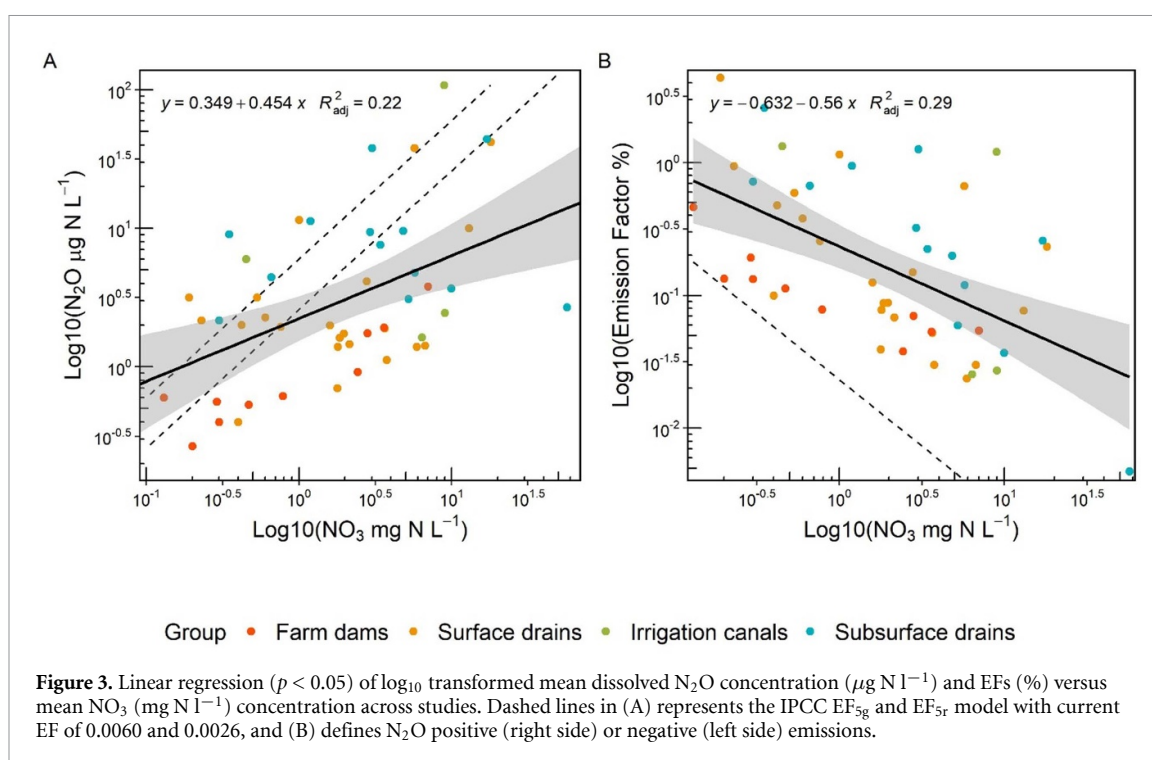


Figure 2. Variation of \log_{10} -transformed dissolved N_2O concentration ($\mu\text{g N l}^{-1}$), NO_3 (mg N l^{-1}) concentrations, and EFs (%) between artificial agricultural surface waters. Different letters represent significant differences ($p < 0.05$) among groups (one-way ANOVA and Tukey–Kramer post hoc test). Boxes span the interquartile range (25%–75% quartiles), whiskers 5%–95% of observations, horizontal line the median, and circle points represent outliers.

3.3. Relationships between N_2O concentration and other factors

Regression analysis with log-transformed mean N_2O concentrations revealed significant relationships with the $\text{DOC}:\text{NO}_3$ ratio, pH, and annual precipitation (figure S4), although these were limited by the number of studies that reported them. Lower N_2O concentrations were observed with either an increase in the $\text{DOC}:\text{NO}_3$ ratio ($R^2 = 0.2$, $p < 0.05$) or water pH ($R^2 = 0.24$, $p < 0.05$), while a weaker correlation ($R^2 = 0.07$, $p < 0.05$) with annual precipitation



showed an increase in N_2O with higher precipitation. No significant relationship was found between N_2O concentrations and N inputs from fertilizer and manure.

4. Discussion

Significant differences in N_2O concentrations were observed between artificial waters, yet the large variability in NO_3 concentrations resulted in no distinct difference in EFs (figure 2). Seasonal variability presents a limitation in the current dataset analysis, with some studies reporting temporally resolved means and while others account for spatial variability (e.g. Webb *et al* 2019a), which could influence the lack of difference observed between some water types here. Given only ~50% of the dataset included seasonal measurements, future studies should examine seasonal impacts on EFs, as water temperatures affects gas dissolution and rainfall/drought events can influence NO_3 and substrate loadings (White *et al* 2021).

As reported in other studies and in the current dataset, N_2O concentrations do not linearly increase with increasing NO_3 concentration, which is the assumption taken when using stationary EFs for estimating indirect emission. This assumption does not take into account the different conditions that drive rates of N_2O production and consumption within each water type. The significant differences in N_2O but not NO_3 concentrations observed between subsurface drainage, surface channels, and farm dams supports the idea of mechanistic differences in N_2O processes. Here, we summarise the data and current understanding of mechanisms driving

N_2O production in each artificial water type explored in this review.

4.1. Subsurface drains

Sub-surface drainage is used to protect soils from waterlogging and manage salinity in agriculture, and in some regions can occur across 80% of the cultivated landscape, often for annual row-crop production and perennial tree crops (Dinnes *et al* 2002). Sub-surface drains exist in tandem with surface drains as part of an integrated drainage system, including piped drains, field ditches, collection ditches, and an outlet drain. The global extent of subsurface drainage is unknown, however, it occurs in 10%–100% of agricultural soils across European countries and ~25% in Northern America (Herzon and Helenius 2008). In this review, 15 studies had reported sub-surface drainage N_2O concentration measurements, installed at depths ranging from 0.5 to 1.6 m below the surface. Livestock agriculture was the main land use reported in addition to some mixed arable production. The majority of sites were located in the UK ($n = 9$), with the remaining from Japan (3), Canada (2), and Denmark (1) (figure S1). In almost all the studies, water was collected at the outlet of the subsurface drainage systems and, as a result, likely underestimated the true EF value due to rapid degassing of N_2O . This is one of the major limitations of sampling subsurface field drainage and requires consideration in future measurements (Roper *et al* 2013).

The mean EF of 0.52% for subsurface drains was significantly higher than other artificial water groups, and is comparable to the IPCC EF_{sg} recommendation of 0.60% for groundwater and drainage waters. The

higher N_2O concentrations observed within subsurface drains may be attributed to a number of factors (figure 2). Firstly, the key difference between these artificial waters with the others reviewed here is their lack of exposure to the open atmosphere, which regulates the rate of N_2O evolution from the water surface. Additionally, *in situ* production and consumption of N_2O is limited due to both the rapid delivery of drainage water once field water enters the drainage system and lack of bottom substrate to support microbial activity. If tile drains do develop any biofilm area, short water residence times may preclude any significant NO_3 removal. Furthermore, supersaturated N_2O concentrations almost certainly originate from the same water source as shallow groundwater: leached soil pore water (Dowdell *et al* 1979). Therefore, subsurface drainage likely represents more of a passive pipe system, transporting dissolved N_2O produced in soil with little outgassing until water reaches a drainage outlet.

N_2O concentrations in subsurface drains are more closely linked to terrestrial characteristics, including soil type, land use and fertilizer practices. A UK study of an intensive arable catchment found that soil texture significantly influenced N_2O concentrations and the relationship with other water quality parameters in a subsurface field drain (Hama-Aziz *et al* 2017). Soils with poor drainage may provide more favourable conditions for N_2O production through increased anoxia and contact with the soil matrix (Hénault *et al* 2012, Jamali *et al* 2016), allowing for more N_2O to dissolve into leached soil water. The strong link with terrestrial N cycling processes is further supported by various studies demonstrating how subsurface drainage design can significantly influence soil N attenuation and direct N_2O emissions (Clagnan *et al* 2018).

4.2. Surface drains

Globally, artificial drainage exists in 130–200 million ha of cropland worldwide, and is increasing in extent and intensity (Schultz and De Wrachien 2002, Castellano *et al* 2019). Surface drains are part of the greater artificial drainage system, often co-existing with subsurface drains, yet are responsible for delivering eutrophying nutrients to natural aquatic and coastal ecosystems (Blann *et al* 2009). Here, 22 studies reported N_2O – N/NO_3 – N concentration and EFs in surface drains, with water depths ranging from 0.06 to 0.6 m. Mixed arable production was the predominant land use followed by livestock, rice, and a palm oil plantation (supplementary data 1). Around 50% of studies were located in the UK, with others distributed across Europe, China, Japan, Indonesia, US, and New Zealand (figure S1). Mean EF was 0.46% (0.02%–4.37%), which is lower than the IPCC EF_{5g} recommendation of 0.60% for groundwater and drainage waters.

Drain position and location within the network are likely important variables to consider when estimating N_2O EFs. Surface drains had significantly lower N_2O concentrations compared to subsurface drains, which supports observations in studies measuring surface drains and tile drains in connected artificial drainage systems (Sawamoto *et al* 2002, Reay *et al* 2004a). Variable physical gas transfer, discharge rate, dilution effects, and distance from point sources can substantially influence within-drain N_2O variability (Sawamoto *et al* 2002, Reay *et al* 2004a, Premaratne *et al* 2017). For example, open surface drains can support large atmospheric N_2O losses during the transport of field water, with dissolved N_2O reductions up to 80% (Reay *et al* 2003, 2004b). This is a result of the relatively high gas transfer velocity generated by field drain outfalls, weirs, and changes in discharge velocity based on the size or shape of a drain. As such, large changes in N_2O concentrations relative to NO_3 concentrations can occur over small spatial scales which leads to large variability in EFs.

Adding to their complexity, some surface drains may receive water from a variety of land uses and sources, or have a more established benthic structure or vegetation environment dependent on stream hierarchy and channel morphology. Headwater agricultural channels can have exponentially higher N_2O emissions than higher-order waterways due to higher N loading, higher N processing rates, and less water volume dilution effects (Turner *et al* 2015). Headwater channels typically receive substantial sediment yields from agricultural fields, resulting in sediment storage with accumulation rates as high as $\sim 90 \text{ kg m}^{-1} \text{ yr}^{-1}$ (Lecce *et al* 2006). This benthic environment can help attenuate agricultural nutrient loads through denitrification at the sediment–water interface, subsequently removing NO_3 from the water column (Veraart *et al* 2017). The current assumption for these drainage systems within the IPCC EF_{5g} model is that NO_3 and N_2O are sourced from groundwater with little in-stream nitrification–denitrification. However, studies have shown strong denitrification activity in drainage ditches across a variety of land use and soil types (Veraart *et al* 2017, Soana *et al* 2019).

4.3. Irrigation canals

Globally, irrigation covers an area of over 300 million ha, which has doubled in the last 50 years and is expected to expand in developing countries (Schultz and De Wrachien 2002, Neumann *et al* 2011). Irrigation farming systems have some of the largest agricultural N losses from hydrological pathways, up to 46%–76% of total N applied to fields (Thorburn *et al* 2011, Perego *et al* 2012). Therefore, there is a need to characterise N_2O emissions from these dynamic systems, however, findings from this review reveal a severe lack of empirical data in this area.

Of the artificial waters collected in this review, only four recorded dissolved N_2O measurements within irrigation waters (table S1). The studies were distributed in the lower latitude regions including Japan, Australia, and Mexico where cotton, rice, and mixed livestock-rice were the main production types. Reported water depths ranged from 0.03 to 1 m. Irrigated farming systems are most prominent in semi-arid regions, with two thirds located in Asia (Molden 2007). Irrigation canals had a mean EF of 0.65% (0.03%–1.33%), which is comparable to the IPCC EF_{5g} recommendation of 0.60%.

However, with such a limited number of studies, it remains difficult to draw conclusions on the difference in N_2O production from irrigation waters with other artificial waters. A large N_2O concentration range of 1.6–108 $\mu\text{g N l}^{-1}$ in canals led to no significant difference when compared to subsurface and surface drains (figure 2). Although not shown here, irrigation canals have some key functional differences with surface drainage channels that may give rise to different N_2O emission estimates if the database for these systems increases. Field runoff collected by canals is largely driven by irrigation events rather than rainfall (Harrison *et al* 2005). Further, irrigation canals can sometimes be free of both macrobenthic organisms and benthic macrophytes due to regular maintenance and concrete construction in some high value farming regions (Kitamura and Nakaya 2010). Finally, many on-farm irrigation canals undergo regular wetting and drying cycles with the irrigation regime, or remain flooded but have highly fluctuating water tables, which exposes bottom sediments or channel banks to the atmosphere. The indirect N_2O emissions released from the exposed sediment phase remains an area needing further investigation (Schwenke *et al* 2020).

The timing and length of irrigation events controls the supply of N from soils to the irrigation water and this will influence N_2O levels within irrigation canals collecting irrigation drainage. For example, an Australian cotton farm irrigation study speculated that low N_2O concentrations and EFs observed within the on-farm irrigation channels were due to a soil water deficit in preceding irrigation events and the short surface water application time (Macdonald *et al* 2016). In addition, field water contact with groundwater may further alter the ratio of N_2O to NO_3 . Comparing two opposing studies reporting either no groundwater contact (Macdonald *et al* 2016) or contact with groundwater (Hasegawa *et al* 2000), the mean EF is two orders of magnitude higher in the groundwater-fed irrigation system. The mixed livestock and rice farm in Japan was an extreme case in terms of N_2O concentrations (7–407 $\mu\text{g N}_2\text{O-N l}^{-1}$) within the irrigation group. In this case, high NO_3 groundwater from stock breeding areas flowed from altitude into rice paddies with high organic matter and fertiliser N inputs, which stimulated excessive

N_2O production via denitrification in the receiving irrigation canal (Hasegawa *et al* 2000).

4.4. Farm dams

On-farm dams are ubiquitous within the agricultural landscape, with several million across the globe used for water supply, irrigation, runoff control, and wastewater management (Verstraeten and Poesen 2000). They are highly abundant in agricultural regions within China, the North American Great Plains, and Australia, where densities often exceed 5 per km^2 (Renwick *et al* 2006, Grinham *et al* 2018, Chen *et al* 2019). Although agricultural dams have come into widespread existence since the 1940s (Renwick *et al* 2006), research into the role of these small artificial lentic systems in GHG cycles has only very recently materialized (Grinham *et al* 2018, Ollivier *et al* 2019, Webb *et al* 2019b). Here, 11 studies with N_2O concentration measurements were collected. Water depth ranged from 0.18 to 5.1 m and land use included irrigated crops, mixed arable, rice, pastures, and palm oil plantations. Studies were distributed across China, Australia, UK, France, Canada, and Indonesia (figure S1). Overall, a mean EF of 0.13% (0.04%–0.46%) was found, which is half the IPCC EF_{5r} estimate of 0.26% for which these systems are included. Instead, the agricultural pond EF found here is comparable to the EF for lakes, ponds, and reservoirs of 0.12% found in a previous review for the refinement of IPCC guidelines (Tian *et al* 2019). However, despite a significantly lower EF estimate compared to rivers (0.30%), lakes, ponds, and reservoirs were not assigned their own EF in the current IPCC EF₅ guidelines.

In terms of N_2O concentrations, farm dams were significantly lower than all other artificial waters reviewed here (figure 2), reflecting findings found from a limited number of studies comparing different surface waters (Outram and Hiscock 2012, Xia *et al* 2013, Xiao *et al* 2019). Farm dams also had the only significant relationship with NO_3 out of the artificial waters (figure S2), where N_2O increased on a logarithmic scale with increasing NO_3 , suggesting a closer alignment to steady state N processing conditions. These differences may be because farm dams are less hydrologically dynamic compared with drainage water bodies due to their role in water storage. Higher water residence times allow for more time for runoff N to be transformed into N_2O , NO_x or N_2 and can even be favourable for N_2O consumption processes (Webb *et al* 2019a). For example, the development of strong thermoclines under steady hydrological conditions has been shown to be a strong influencing factor in supporting dissolved N_2O undersaturation (Webb *et al* 2019a). Farm dams also have a lower gas transfer velocity compared to channels (Premaratne *et al* 2017, Ollivier *et al* 2019, Webb *et al* 2019a), leading to slower gas exchange between the water–atmosphere surface and potentially less

discrepancy between changes in N_2O and NO_3 concentrations.

Introducing on-farm dams into intensive agricultural landscapes, especially those with a high density of tile drainage, may provide a measure to reduce potential N_2O emissions further downstream. The strong reductive conditions enhanced due to low water velocity enables ponds to remove greater amounts of N than streams (Li *et al* 2013, Garnier *et al* 2014). This supports their ability to receive high inorganic N loads without the consequence of producing proportional N_2O emissions (Webb *et al* 2019a). The high density of streams and ponds scattered within the rice-paddy-dominated watersheds, characteristic of China, is an example of where high N input does not always lead to high N concentrations in the surface water (Xia *et al* 2013, Xiong *et al* 2015). Further, a study of the Orgeval watershed in France explored the role of pond implementation in drainage areas and predicted a 34%–47% reduction in surface water N export (Passy *et al* 2012).

4.5. Low EF in waters with high N loading

Regression analysis between mean N_2O concentrations and NO_3 concentrations revealed a significant positive-logarithmic rather than a linear response, and that in most cases $\text{EF}_{5\text{g}}$ and $\text{EF}_{5\text{r}}$ models overestimated N_2O concentrations for artificial waters (figure 3(A)). A stronger relationship between NO_3 and EF was found where EF scales inversely with higher NO_3 (figure 3(B)). This relationship was consistent across subsurface drains, surface drains, and farm dams (figure S3), and suggests that N_2O emissions in most artificial waters are overestimated using standard EF_5 modelling. Non-linearity in the N_2O concentration to NO_3 response has been observed in rivers, implying a limit on NO_3 processing and subsequent N_2O production (Turner *et al* 2016, Wang *et al* 2018). Moving away from stationary EFs and developing a model that accounts for the inverse scaling of EF with higher N loading will greatly reduce N_2O emission uncertainties from highly N polluted agricultural waters.

This pattern of lower N_2O production with increasing NO_3 loading is consistent with data from previous studies, although the processes driving this trend remains unclear. In agricultural watersheds receiving high N loads, less N_2O may be produced relative to NO_3 due to biological saturation (Mulholland *et al* 2008, Beaulieu *et al* 2011, Xiao *et al* 2019). Alternatively, undersaturation of dissolved N_2O in the presence of high NO_3 can occur if the right reductive conditions exist to facilitate complete denitrification, as seen in farm dams in Canada (Webb *et al* 2019a). Furthermore, in open water systems any N_2O produced in excess of atmospheric equilibrium will degas into the atmosphere more rapidly than microbial NO_3 transformations. If much of the N_2O has already degassed, then discrepancies in the EF_5 method that

derives ratios between actual measured fluxes of N_2O and NO_3 versus dissolved concentrations will arise (Clough *et al* 2006).

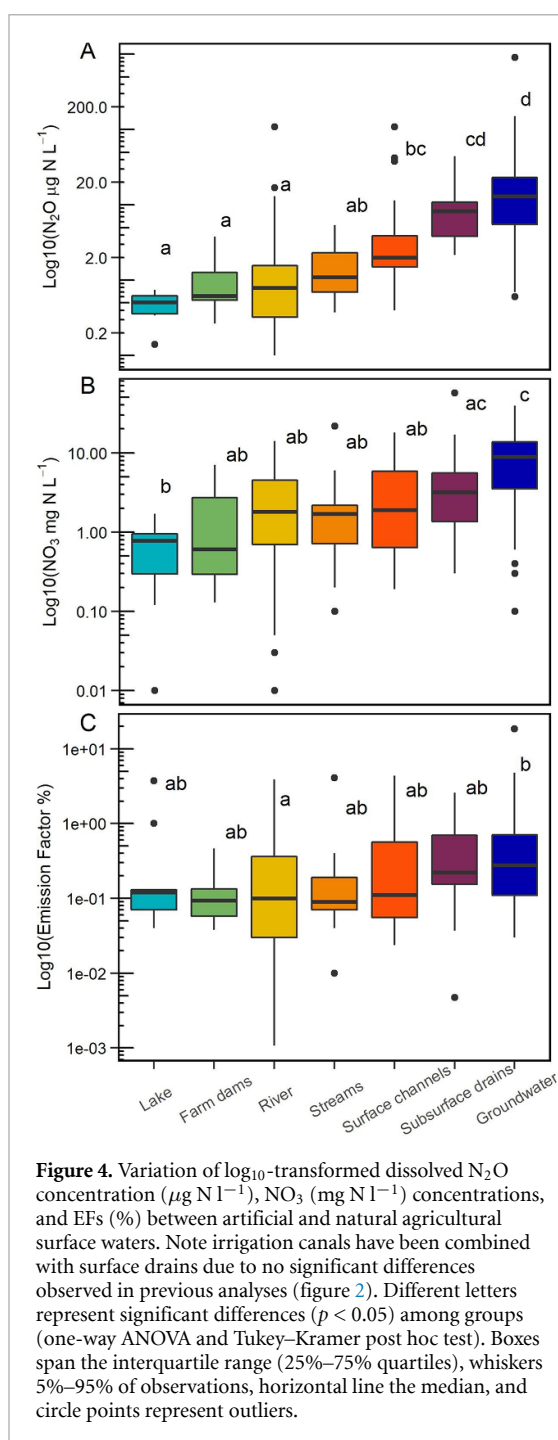
4.6. Artificial and natural waters in emission factor groups

To determine if artificial waters deserve recognition separate to natural waters for indirect N_2O emission accounting, we compared N_2O concentrations, NO_3 , and EFs from artificial waters collected in this study with natural waters contributing to the current $\text{EF}_{5\text{g}}$ and $\text{EF}_{5\text{r}}$ models. In this analysis, irrigation canals were included with the surface drainage group, renamed to surface channels, due to the small number of studies and the fact that no significance difference was detected in previous analyses (figure 2). Natural waters were divided into groundwater, lakes, rivers, and streams as defined in Tian *et al* (2019). Despite finding significant differences in N_2O concentrations, findings revealed no significant differences in EFs between artificial and natural waters (figure 4). This suggests that delineation of water groups into artificial and natural waters is unlikely to improve EF uncertainty within the current IPCC framework for assigning default EFs.

Mean N_2O concentrations in farm dams were not significantly different between lakes, rivers, and streams, yet surface channels were significantly different to groundwater. Notable differences governing N_2O concentrations in response to N loadings from artificial systems compared to natural waters have been found within agricultural watersheds. This included river N_2O dynamics being more affected by heavy rainfall events than rice paddy ponds (Xiao *et al* 2019), and differences in redox conditions and wind turbulence affecting N_2O production between ditches and a river (Outram and Hiscock 2012). In contrast, both N_2O and NO_3 concentrations for subsurface drainage were comparable to groundwater (figure 4), supporting the assumption that leached soil pore water is the primary source of supersaturated N_2O in both waters (Well *et al* 2005). On this basis, the IPCC EF grouping for $\text{EF}_{5\text{g}}$ is likely correct to include groundwater and subsurface drainage together, but not open water surface drainage (figure 4).

4.7. Limitations and future research

A number of competing factors that alter N_2O and NO_3 concentrations, either separately or collectively, complicates the assumed linear response when estimating N_2O emissions using default EFs. Some of these factors are more apparent in different artificial waters, such as higher gaseous losses in surface channels and where there is a greater capacity for N_2O consumption, such as farm dams. As others have proven, a process-based model instead of revising default EFs will likely lead to the most noticeable improvements in estimating indirect aquatic N_2O emissions



(Maavara *et al* 2019). However, the ability to develop process-based models for specific water types is limited by a severe lack of studies on artificial waters. Until then, a number of limitations with the current model of predicting N_2O emissions needs to be considered.

Firstly, lack of consistent reporting of other important environmental factors that influence N_2O production impedes development towards a process-based model. For example, proximity of open channels to agricultural fields may play a role in driving EFs, as stream order and distance from the N source have been shown to significantly impact N_2O concentrations (Reay *et al* 2004a; Turner *et al* 2015). Many

of the environmental variables have been found to drive N_2O concentrations in artificial waters, including the $\text{DOC}:\text{NO}_3$ ratio, pH, fertilizer application, and precipitation, were tested across a subset of the artificial waters collected in this review (figure S4). The N_2O concentration was inversely related to both $\text{DOC}:\text{NO}_3$ and pH, while annual precipitation was less strong as a predictor and no relationship was found with total N application. Variables such as DOC and pH are often found to be strong drivers of aquatic N_2O dynamics, as at higher values they can represent conditions more favourable to microbial N reduction (Peacock *et al* 2017, Audet *et al* 2020). However, values for these factors could only be derived from 30% to 40% of studies reviewed here.

The large range in EFs within groups contributes to major uncertainty when applying default values to local sites. Grouping artificial waters into their respective groups also revealed a large range in EF, as well as a highly skewed distribution in N_2O , NO_3 , and EF data (figure S5). Bias can arise when upscaling emissions using the mean statistic when the mean and median of a dataset are distinctly different, such as in surface drains with a mean and median EF of 0.46% and 0.11%, respectively (table S1). As discussed by others, care must be taken when extrapolating means derived from a skewed dataset to estimate GHG emissions (Grinham *et al* 2018; Rosentreter and Williamson 2020). Due to the low number of observations and geographical bias of studies collected here, the N_2O , NO_3 , and EF datasets are unlikely to represent the ‘real-world’ distribution. In fact, ~50% of studies were located in the UK, which represents a notable limitation and a potentially sizeable source of error when extrapolating EFs globally. The choice of the arithmetic mean statistic in these cases may not provide a representative estimate for local-scale N_2O emissions and may explain why the IPCC default EFs are often found to be overestimated.

Default EFs that rely on the mass ratio of N_2O to NO_3 concentrations are also complicated by kinetic limitations introduced by variable gas exchange rates and water residence times across different water types. Open drains in particular represent complex aquatic systems for constraining N_2O emissions due to their dynamic hydrology and variable design. Where rapid gaseous N_2O losses occur within a water body, the $\text{N}_2\text{O}:\text{NO}_3$ ratio will be smaller as the biological processes consuming NO_3 operate at a slower time scale than physical evasion to the atmosphere. Additionally, water residence time introduces a hydrological constraint on the extent of in-system N transformations (Maavara *et al* 2019). Together, the physical constraints from gas transfer and hydrologic exchange result in a variable $\text{N}_2\text{O}-\text{N}/\text{NO}_3-\text{N}$ concentration ratio that alters biological processing rates across different water types.

The IPCC provides two different methods for calculating indirect N_2O EFs, as presented in the

introduction, and it is important to be aware of the different EF values these two methods may produce. There can be large discrepancy between EFs estimated from mass N-fluxes and EFs estimated from N-concentrations ratios (as reviewed here), as reported in Clough *et al* (2006) and Outram and Hiscock (2012), whereas at other times the two methods yield similar estimates (e.g. Hama-Aziz *et al* 2017, Premaratne *et al* 2017). In the few studies that reported EF₅ derived from both methods, or provided detailed information to allow the calculation, differences in the average EF ratio ranged from 0.00003 to 0.0469 in surface drainage systems and 0.0001 to 0.003 in subsurface drainage (supplementary table S3). Often, N₂O fluxes were calculated using gas exchange models, which can be a large source of uncertainty if not produced specifically for the local water system. Development of models for calculating gas transfer velocity across specific artificial water types will advance the field by enabling standardised emissions upscaling of dissolved N₂O concentrations. Some progress has been made by recent studies in this area, which report gas transfer coefficients of 3.8–6.6 m d⁻¹ for surface drains (Premaratne *et al* 2017) and 1–1.64 m d⁻¹ for on-farm dams (Ollivier *et al* 2019, Webb *et al* 2019).

A hybrid modelling approach that integrates the biological and chemical controls of N₂O process rates with hydromorphological properties (e.g. Marzadri *et al* 2020) of artificial waters, including their type, area, water residence time, and distance from farm fields or runoff source, is likely required to capture the complexity of these systems. However, until these local-scale processes for artificial waters can be up-scaled in modelling efforts, scaling EFs with NO₃ loading may prove to be a promising first step to move away from the default EF approach. Significant inverse correlations were found across all water groups (excluding irrigation canals) between EF and NO₃, while only farm dams revealed NO₃ to be a significant driver of N₂O (figure S3). To develop this concept further, future research should explore N₂O production thresholds within high NO₃ environments to better understand the conditions that lead to the inverse EF to NO₃ relationship. Some studies have started progressing this area, by investigating denitrification rates/changes in ditches receiving greater NO₃ inputs (Veraart *et al* 2017, She *et al* 2018), and looking at stratification in farm dams supporting N₂O consumption under high NO₃ conditions (Webb *et al* 2019a). Another variable worth investigating is the DOC/NO₃ ratio, which we also found to be a significant inverse driver of N₂O in artificial waters, as DOC/NO₃ can be a proxy for N limitation within aquatic ecosystems (Peacock *et al* 2017).

Finally, it is difficult to assess the relative impact of artificial waters on regional and global indirect N₂O emissions until a database exists on the area and distribution these systems cover. The recent

revised quantification of global N₂O sources and sinks attributes 0.5 Tg N yr⁻¹ to indirect anthropogenic emissions from streams, rivers, lakes, reservoirs (>0.1 km²), and estuaries, which is equivalent to 13% of total direct agricultural N₂O emissions. However, this estimate does not include small artificial waters. This gap presents an opportunity to refine the contribution of anthropogenic N₂O emissions in total global land emissions, where an estimated discrepancy of ~1.8 Tg N yr⁻¹ exists between bottom-up and top-down land models (Tian *et al* 2020). Explicitly including artificial waters in models for budget quantification will likely make a significant contribution to the agricultural N₂O budget in regions where artificial waterbodies have been densely created, such as the irrigation networks in the Mediterranean region of Spain and irrigation ponds in southern China (Aguilera *et al* 2019, Chen *et al* 2019). Development of national inventories are already underway in some regions (Aguilera *et al* 2019, Malerba *et al* 2021), yet a collective country-by-country effort is needed to quantify the global extent.

5. Conclusion

Our analysis of the available literature suggests that subsurface drains, surface drains, and on-farm dams function differently in terms of surface water N₂O production, while more studies are urgently needed to assess the role of irrigation canals. Studies showed that EFs vary largely in space and time within surface waters, which hinders detection of differences between artificial waters. The huge variability of EFs within water groups challenges the concept that N₂O production in downstream waters is proportional to agricultural NO₃ load. The current IPCC methodology using generalised EFs overestimates indirect N₂O production in waters receiving high NO₃ loading; a characteristic of many artificial waters. In fact, in these systems EFs scale downward with increasing N load. Improving the prediction of indirect N₂O emissions from agricultural surface waters likely requires a move away from stationary EFs and explicit inclusion of different artificial waters. On a global scale, the lack of accounting for artificial waters in the most recent global N₂O budget presents a significant gap in the quantification of indirect agricultural N₂O emissions from aquatic systems (Tian *et al* 2020). As such, we suggest a need for a global database on the size, density, and distribution of artificial waters in agricultural landscapes, as well as a need to expand the global artificial waters N₂O dataset beyond the UK. Further, future field measurements should target different artificial waters across major agricultural industries to advance the biogeochemical understanding to the level that has been achieved for rivers and lakes (Lauerwald *et al* 2019, Maavara *et al* 2019).

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

Acknowledgments

JW acknowledges support by a Deakin University SEBE Industry Postdoctoral Fellow Scheme and the Australian Department of Agriculture, Water and the Environment, Round 1 (2017–2022) National Landcare Programme's Smart Farming Partnerships project: 'Unlocking the true value of soil organic amendments'. Thanks to Michael Peacock, Ollivier, Quinn, and Joachim Audet for providing data, and to Michael Peacock for helpful discussions around artificial waters.

ORCID iD

Jackie R Webb  <https://orcid.org/0000-0003-1274-363X>

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